

Barry, Chester

From: Barry, Chester
Sent: Friday, September 05, 2003 8:13 AM
To: STIC-EIC1700
Subject: 10/866145

pls conduct a litigation search of usp 5906750 thx

Chester T. Barry
Primary Examiner, GAU 1724
United States Patent & Trademark Office
703-306-5921



STIC Search Report

EIC 1700

STIC Database Tracking Number: 103123

TO: Chester Barry
Location: CP3 5D29
Art Unit : 1724
September 5, 2003

Case Serial Number: 10/866145

From: Kendra Mellerson
Location: EIC 1700
CP3/4-3D62
Phone:308-4483

Kendra.Mellerson@uspto.gov

Search Notes

No Cases Reported

Current session 05/09/2003

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QUESTEL.ORBIT (TM) 1998

05/09/03 15*11*58

Last connection: 03/09/03 15*25*23

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QUESTEL - Time in minutes : 0,65

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Last update of file: 2003/09/04 (YYYY/MM/DD) 2003-35/UP (basic update)

Search statement 1

Query/Command : US5906750/PN

** SS 1: Results 1

Search statement 2

Query/Command : PRT FULL NONSTOP LEGALALL

1/1 PLUSPAT - ©QUESTEL-ORBIT

PN	-	US5906750 A 19990525 [US5906750]
TI	-	(A) Method for dewatering of sludge
IN	-	(A) HAASE RICHARD ALAN (US)

AP - US5587098 19980406 [1998US-0055870]
FD - C.I.P. of US721557 19960926 [1996US-0721557]
Continuation-in-part of: US5846435
PR - US5587098 19980406 [1998US-0055870]
US72155796 19960926 [1996US-0721557]
IC - (A) C02F-011/12
EC - C02F-011/14
PCL - ORIGINAL (O) : 210727000; CROSS-REFERENCE (X) : 210609000 210728000
DT - Basic
CT - US3994806; US4784776; US4882069; US4996706; US5093078; US5112603; US517877
US5366637; US5529700; US5643462; US5681480
STG - (A) United States patent
AB - A chemical method is provided for the dewatering of biological sludge that has been digested by a thermophilic digestion process. Five versions of the chemical sludge dewatering method are presented. The primary component in the five versions is a polyquaternary amine, preferably the di-allyl di-methyl ammonium chloride (DADMAC) variety and from the epichlorohydrin dimethyl amine (epi-DMA) variety. By the first sludge dewatering method, the polyquaternary amine is added directly, along with a polyacrylamide, to the biological sludge. By the second sludge dewatering method, the polyquaternary amine and an anionic polyacrylamide are added separately. By the third sludge dewatering method, a quaternized polyacrylamide, having the polyquaternary amine as part of its polymer chain, is produced by copolymerization of acrylamide with monomers of polyquaternary amine quaternization and is added individually to the sludge. By the fourth sludge dewatering method, the quaternized polyacrylamide from method three is added in concert with a cationic polyacrylamide to the sludge. By the fifth dewatering method, aluminum sulfate, ferric chloride and blends thereof are added, along with polyquaternary amine, as the primary component to the sludge.

1 / 1 LGST - ©LEGSTAT

PN - US 5906750 [US5906750]
AP - US 55870/98 19980406 [1998US-0055870]
DT - US-P
ACT - 19980406 US/AE-A
APPLICATION DATA (PATENT)
US 55870/98 19980406 [1998US-0055870]

19990525 US/A
PATENT

20011120 US/RF
REISSUE APPLICATION FILED
20010525
UP - 2001-48

1 / 1 CRXX - ©CLAIMS/RRX

PN - 5,906,750 A 19990525 [US5906750]

PA - Haase, Richard Alan
ACT - 20010525 REISSUE REQUESTED
ISSUE DATE OF O.G.: 20011120
REISSUE REQUEST NUMBER: 09/866145
EXAMINATION GROUP RESPONSIBLE FOR REISSUEPROCESS: 1724

Reissue Patent Number:

Query/Command : FILE INPADOC

PLUSPAT - Time in minutes : 0,38
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LGST - Time in minutes : 0,11
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Estimated cost :	0.13 USD
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Legal-Status informations :	1
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LITA - Time in minutes : 0,01
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Estimated cost :	0.02 USD
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Estimated total session cost :	9.86 USD

Selected file: INPADOC

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Search statement 1

Query/Command : FAM US5906750/PN

1 Patent Groups

** SS 1: Results 4

Search statement 2

Query/Command : FAMSTATE NONSTOP

1 / 4 INPADOC - ©INPADOC

PN - AU 32195/99 A1 19991025 [AU9932195]
TI - METHOD FOR DEWATERING OF SLUDGE
IN - HAASE RICHARD A
PA - RICHARD A HAASE
AP - AU 32195/99-A 19990401 [1999AU-0032195]
PR - US 55870/98-A 19980406 [1998US-0055870]
WO 9907139/99(US)-W 19990401 [1999WO-US07139]
IC - C02F-011/12

1 / 1 LEGALI - ©LEGSTAT

PN - AU 32195/99 [AU9932195]
DT - AU-A
ACTE - 19990401 AU/AE-A
19991025 AU/A1
UP - 2000-05

2 / 4 INPADOC - ©INPADOC

PN - US 5846435 A 19981208 [US5846435]
TI - METHOD FOR DEWATERING OF SLUDGE
IN - HAASE RICHARD ALAN [US]
PA - HAASE RICHARD ALAN [US]
AP - US 721557/96-A 19960926 [1996US-0721557]
PR - US 721557/96-A 19960926 [1996US-0721557]
IC - C02F-011/12

1 / 1 LEGALI - ©LEGSTAT

PN - US 5846435 [US5846435]
AP - US 721557/96 19960926 [1996US-0721557]
DT - US-P

ACTE - 19960926 US/AE-A
APPLICATION DATA (PATENT)
US 721557/96 19960926 [1996US-0721557]

19981208 US/A
PATENT

20000606 US/RR [+]
REQUEST FOR REEXAMINATION FILED
20000424

20010213 US/RF
REISSUE APPLICATION FILED
20001207
UP - 2001-07

3 / 4 INPADOC - ©INPADOC

PN - US 5906750 A 19990525 [US5906750]
TI - METHOD FOR DEWATERING OF SLUDGE
IN - HAASE RICHARD ALAN [US]
PA - HAASE RICHARD ALAN [US]
AP - US 55870/98-A 19980406 [1998US-0055870]
PR - US 55870/98-A 19980406 [1998US-0055870]
US 721557/96-A2 19960926 [1996US-0721557]
IC - C02F-011/12

1 / 1 LEGALI - ©LEGSTAT

PN - US 5906750 [US5906750]
AP - US 55870/98 19980406 [1998US-0055870]
DT - US-P
ACTE - 19980406 US/AE-A
APPLICATION DATA (PATENT)
US 55870/98 19980406 [1998US-0055870]

19990525 US/A
PATENT

20011120 US/RF
REISSUE APPLICATION FILED
20010525
UP - 2001-48

4 / 4 INPADOC - ©INPADOC

PN - WO 9951533 A1 19991014 [WO9951533]
TI - METHOD FOR DEWATERING OF SLUDGE

LA - ENG
IN - HAASE RICHARD A [US]
PA - HAASE RICHARD A [US]
AP - WO 9907139/99(US)-A 19990401 [1999WO-US07139]
PR - US 55870/98-A 19980406 [1998US-0055870]
IC - C02F-011/12
DS - AL* AM* AT* AU* AZ* BA* BB* BG* BR* BY* CA* CH* CN* CU* CZ* DE* DK*
 ES* FI* GB* GD* GE* GH* GM* HR* HU* ID* IL* IN* IS* JP* KE* KG* KP* KR* K
 LC* LK* LR* LS* LT* LU* LV* MD* MG* MK* MN* MW* MX* NO* NZ* PL* PT*
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1/1 LEGALI - ©LEGSTAT

PN - WO 9951533 [WO9951533]
AP - WO 9907139/99(US) 19990401 [1999WO-US07139]
DT - WO-P
ACTE - 19990401 WO/AE-A
 APPLICATION DATA
 WO 9907139/99(US) 19990401 [1999WO-US07139]

19991014 WO/AK-A1 [+]

DESIGNATED STATES CITED IN A PUBLISHED APPLICATION WITH SEARCH REPORT

AL AM AT AU AZ BA BB BG BR BY CA CH CN CU CZ DE DK EE ES FI GB GD GE
 GM HR HU ID IL IN IS JP KE KG KP KR KZ LC LK LR LS LT LU LV MD MG MK M
 MW MX NO NZ PL PT RO RU SD SE SG SI SK SL TJ TM TR TT UA UG US UZ VN
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19991014 WO/AL-A1 [+]

DESIGNATED COUNTRIES FOR REGIONAL PATENTS CITED IN A PUBLISHED APPLICATION WITH SEARCH REPORT

GH GM KE LS MW SD SL SZ UG ZW AM AZ BY KG KZ MD RU TJ TM AT BE CH
 DE DK ES FI FR GB GR IE IT LU MC NL PT SE BF BJ CF CG CI CM GA GN GW ML
 NE SN TD TG

19991014 WO/A1 [+]

PUBLICATION OF THE INTERNATIONAL APPLICATION WITH THE INTERNATIONAL SEARCH REPORT

19991208 WO/121

EP: THE EPO HAS BEEN INFORMED BY WIPO THAT EP WAS DESIGNATED IN THIS APPLICATION

20000810 WO/REG; DE/8642 [-]

DE: IMPACT ABOLISHED FOR DE
 <DE>

20001106 WO/NENP
NON-ENTRY INTO THE NATIONAL PHASE IN:
<RU>

20001227 WO/122 [-]
EP: PCT APP. NOT ENT. EUROP. PHASE

UP - 2003-06

PATNO IS 5906750

DATE: SEPTEMBER 5, 2003
LIBRARY: PATENT
FILE: ALL

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PATNO IS 5906750

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To MODIFY your search request, press the M key (for MODIFY) and then the ENTER key.

For further explanation, press the H key (for HELP) and then the ENTER key.

LEVEL 1 - 1 PATENT

1. 5906750 , May 25, 1999 , Method for dewatering of sludge, Haase, Richard
Alan, P.O. Box 623 Sugar Land, TX 77487-0623, 055870 (00)

LEVEL 1 - 1 OF 1 PATENT

UNITED STATES PATENT AND TRADEMARK OFFICE GRANTED PATENT

5906750

May 25, 1999

Method for dewatering of sludge

REISSUE: Reissue Application filed May 25, 2001 (O.G. Nov. 20, 2001) Ex. Gp.:
1724; Re. S.N. 09/866,145, (O.G. November 20, 2001)

APPL-NO: 055870 (00)

FILED-DATE: April 6, 1998

GRANTED-DATE: May 25, 1999

5906750 OR 5,906,750

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2	15	((("5137599") or ("5965027") or ("4380496") or ("3613564") or ("3397139") or ("6083404") or ("4396513")).PN.	USPAT; US-PGPUB; EPO; JPO; DERWENT; IBM_TDB	2003/09/05 11:37
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4	22	gregor.in. and 210/\$.ccls.	USPAT; US-PGPUB; EPO; JPO; DERWENT; IBM_TDB	2003/09/05 11:39
5	2	5906750.pn.	USPAT; US-PGPUB; EPO; JPO; DERWENT; IBM_TDB	2003/09/05 12:15
6	11	("3994806" "4784776" "4882069" "4996706" "5093078" "5112603" "5178774" "5366637" "5529700" "5643462" "5681480").PN.	USPAT	2003/09/05 11:39
7	2	4966706.pn.	USPAT; US-PGPUB; EPO; JPO; DERWENT; IBM_TDB	2003/09/05 11:40
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9	1	5906750.pn. and "primary sludge"	USPAT; US-PGPUB; EPO; JPO; DERWENT; IBM_TDB	2003/09/05 12:52
10	84	epi-dma	USPAT; US-PGPUB; EPO; JPO; DERWENT; IBM_TDB	2003/09/05 12:52
11	646	dadmac	USPAT; US-PGPUB; EPO; JPO; DERWENT; IBM_TDB	2003/09/05 12:52
12	372	"primary sludge"	USPAT; US-PGPUB; EPO; JPO; DERWENT; IBM_TDB	2003/09/05 12:52
13	4860	thermophillic or thermophilic or thermophile or thermophyllic	USPAT; US-PGPUB; EPO; JPO; DERWENT; IBM_TDB	2003/09/05 12:53

14	2	epi-dma and dadmac and "primary sludge" and (thermophillic or thermophilic or thermophile or thermophyllic)	USPAT; US-PGPUB; EPO; JPO; DERWENT; IBM_TDB	2003/09/05 12:53
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Mellerson, Kendra

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Subject: 10/866145

pls conduct a litigation search of usp 5906750 thx

Chester T. Barry
Primary Examiner, GAU 1724
United States Patent & Trademark Office
703-306-5921

Evaluation of Dual Chemical Conditioning and Dewatering of Anaerobically Digested Biosolids

**The Final Project Report:
Sludge Dewaterability Assessment for East Bay Municipal Utility District
(EBMUD)
California**

**Steven K. Dentel and Srinivasarao Chitikela
Department of Civil and Environmental Engineering
University of Delaware
Newark, DE 19716**

June, 1995

Introduction

The optimal chemical conditioning and dewatering of a municipal sludge is a challenging task. This is due to the complex and changeable physical/chemical makeup of these materials. The means by which chemical conditioners interact with the colloidal phase in biological suspensions to facilitate the release of water is poorly understood, with the optimal amounts and types of conditioners required depending on a variety of factors. These include both aqueous and surface chemistries of the sludge, and the physical properties of the suspended solids, which are determined by characteristics of the original wastewater and by the operational parameters for the various treatment processes employed in the plant. Also important is the chemistry of any chemical conditioner used, and how it interacts with the biosolids. The success of any conditioning process will also depend on the specific dewatering process employed.

Thus, the conditioning process is a multivariate problem with no simple strategy available for its optimization. At present, the required dosages for chemical conditioners must be determined empirically. With this being the case, the use of multiple chemical additives becomes less feasible because of the difficulty in identifying a proper dose combination. Nonetheless, it is possible that a combination of conditioners might enhance water removal, solids capture, and solids yield for some applications.

This work investigated the combined use of an inorganic conditioner (ferric chloride) and an organic polyelectrolyte for the conditioning and dewatering of municipal anaerobically digested sludges. In the past, ferric chloride was more commonly used in conjunction with lime, but the current practice is generally to use cationic polymers (polyelectrolytes) alone. The inorganic conditioners require doses up to 20% on the dry solids basis and typically cannot produce the solids concentrations in dewatered biosolids that are attainable with much lower dosages of polymer.

Thus, in spite of their higher unit cost, organic polymers have largely displaced inorganic chemicals in sludge conditioning and dewatering processes. U.S. polymer sales for this purpose are estimated at \$130 million per year (Dentel et al., 1995). The expense of polymer purchase is usually the greatest single cost component in biosolids management, and thus represents a considerable portion of overall treatment costs. At some treatment facilities where polymer demand is unusually high, this expense may even exceed secondary treatment aeration costs.

When polymer costs become a disproportionate financial burden, it may be worthwhile to assess the use of a less expensive inorganic conditioner for at least part of the chemical usage. Ferric chloride is one alternative which is already provided in some facilities for

struvite control. Its potential use in tandem with a polymer for less expensive dewatering was therefore of interest.

This study was initiated by the the East Bay Municipal Utility District (EBMUD) in Oakland, California due to its high polymer usage in conditioning anaerobically digested sludge (ADS). For such applications the EPA (1987) indicates a usual dose range of 3-15 lb/ton with a typical value of 6 lb/ton. The required dose at this facility was approximately 34 lb/ton of dry solids. Previous laboratory comparisons based on capillary suction time (CST) had confirmed the necessity for this high dose: for example, to attain the CST resulting from a 15 lb/ton polymer dose to an ADS from Wilmington, Delaware, the EBMUD biosolids required 34 lb/ton of the same polymer. Both electrical charge and rheological properties of the sludges were indicated as contributing factors (Dentel et al., 1994).

The EBMUD sludge was shown to have a more negative zeta potential and a greater concentration of negative charge than the Wilmington sludge. The conditioning mechanisms attributed to ferric chloride include the formation of positively charged iron hydroxide precipitate, which can then neutralize the negatively charged sludge solids. It thus appeared plausible that this conditioner might be substituted for a more expensive polyelectrolyte in accomplishing a portion of the charge neutralization. The cationic organic polymer might still be needed to bridge the particles and provide some of the desired rheological properties. The detailed principles of inorganic chemical and organic polymer conditioning of sludges are available elsewhere (EPA, 1987; Gregory, 1993; Abu-Orf, 1994).

Objectives. In this work, we evaluated a two-stage conditioning of anaerobically digested sludges consisting of initial addition of ferric chloride followed by use of a cationic polymer. Using capillary suction time (CST) as a laboratory indicator of dewaterability, combined doses were to be identified that produced CST values equal to that obtained with polymer alone. An economic evaluation of dual conditioner use was to be completed to determine whether a cost reduction could be effected by partial use of the less expensive ferric chloride.

It is possible to view the destabilization of colloidal material in ADS as analogous to the coagulation-flocculation process in water treatment, with the first chemical additive primarily intended for neutralization of colloidal surface charge, and the subsequent polymer used for bridging and strengthening of the flocculated mass. In order to ascertain whether charge neutralization is mechanistically important in this manner, we utilized a streaming current detector to observe trends in colloidal charge occurring with the use of

these conditioners. We also employed a cationic surfactant in place of ferric chloride as the initial chemical conditioner, which was expected to neutralize at least some of the negative charge on the initial biosolids, but with an unknown effect on dewaterability.

Procedures

Materials:

In addition to the biosolids from the EBMUD facility, anaerobically digested biosolids were also obtained from Philadelphia's Biosolids Recycling Center for comparison purposes.

Ferric chloride (formula weight 162.21) was obtained from Fisher Scientific (Malvern PA.) and prepared as a 5.029 percent solution for use. The cationic surfactant used as a conditioner was hexadecyltrimethyl ammonium bromide (HDTMA), a quaternary amine cationic surfactant with a formula weight of 364.5. Also obtained from Fisher, it was prepared and used as a 4 percent solution.

Percol 757 was used as a representative cationic polymer. A product of Allied Colloids, Inc., it is supplied as a dry copolymer of acrylamide and quaternized dimethylaminoethylacrylate at a mole ratio of approximately 60%/40%. It was obtained after preparation to solution form in the full-scale facilities of the Wilmington (Delaware) Water Pollution Control Facility, where it is used for sludge conditioning. The solution was determined to have a total solids content of 0.50 percent. A syringe was used to dispense this solution into biosolids samples.

A household blending mixer (Braun Multipractic) and a Phipps & Bird jar test apparatus with standard 1"x 3" paddles were used for mixing of sludges in the conditioning process. A capillary suction time (CST) instrument (Venture Innovations) was used for dewaterability assessments. A streaming current detector (Milton Roy model SC4200) was used for determination of relative colloidal charge. The pH meter used in this work was an Accumet® model 830 (Fisher Scientific).

Methods:

Conditioning of sludges was conducted using the general procedure given in the Guidance Manual for Selection of Polymers in Wastewater Treatment Plants (WERF, 1994).

i) Conditioning and dewatering of sludges by Percol 757, ferric chloride, or HDTMA alone: A representative 500 mL volume of each sludge was placed in a 1000 mL beaker and rapidly mixed with the household hand mixer (Braun Multipractic) for 6-7 seconds

after addition of the specified dose of Percol 757, ferric chloride, or HDTMA. Then using the jar test apparatus, rapid mixing was continued for 20 seconds at approximately 117 RPM, and finally, the suspension was flocculated at approximately 28 RPM for 2 minutes. The suspension's pH, CST and streaming current (SC) values were then measured. All experimental values presented in this report are averages of at least three analyses.

ii) Conditioning and dewatering of sludges by ferric chloride or HDTMA followed by Percol 757: The same procedure was used except that samples were rapidly mixed with the household blender for 5 seconds after addition of the specified dose of ferric chloride or HDTMA, and for an additional 5 seconds following the specified dose of Percol 757.

The above mixing conditions were selected to simulate as closely as possible the flow conditions (velocity gradients and mixing times) from the point of chemical addition up to the centrifuges at the EBMUD facility, as detailed in our previous report to EBMUD. The 'Gt' value, calculated by summing products of velocity gradients and respective residence times, was between 7300 and 9500 depending on the sludge viscosity (which is dependent in turn on polymer dose). This range has been reported as optimal in previous studies of ADS conditioning (Novak 1989, 1990).

Results and Discussion

The characteristics of the anaerobically digested sludges used in this work are shown in Table 1.

Table 1: Initial characteristics of the sludges used in this study.

Original suspension	Sample pH	Total Solids (percent)	CST (seconds)	Streaming current
ADS from EBMUD	7.5-7.6	2.15	494	-42.2 to -42.8
ADS from Philadelphia	7.5-7.75	2.46	434	-33.8 to -35.0

ADS = Anaerobically digested sludge

A comparison of EBMUD and Philadelphia sludge total solids and pH values shows no major differences. The Philadelphia sample had a slightly greater TS content, by a factor of 14%. All else being equal, the greater solids content of the Philadelphia sludge would lead to a 14% greater polymer demand, since required polymer dosages are generally considered to be proportional to solids levels.

Figures 1 and 2 show the results of conditioning and dewatering results for both the EBMUD and Philadelphia sludges, when conditioned with Percol 757, ferric chloride, or HDTMA individually. These results once again confirm that cationic polymers are very effective in sludge conditioning when compared to the inorganic chemical conditioning with ferric chloride. Use of a cationic surfactant in sludge conditioning is unconventional, but its optimum dose on a mass basis was comparable to that of ferric chloride. However, this finding is of little practical use since HDTMA is significantly more expensive than ferric chloride or flocculant polymers per unit mass. Cationic surfactants such as HDTMA are also used as germicides, herbicides, fabric softeners, and corrosion inhibitors (Rosen, 1989) and thus would require evaluation with respect to other impacts as well.

These results also show a greater conditioner requirement for the EBMUD biosolids than for the Philadelphia biosolids. To obtain a CST of approximately 10 seconds, the required Percol 757 additions for EBMUD and Philadelphia respectively were 215 and 180 mg/L or, on a mass per mass basis, 20 and 15 lb/ton. The ferric chloride additions never reduced the CST to 10 seconds for either sample, but to provide a CST of 15 seconds the doses were 6000 and 5200 mg/L respectively (560 and 430 lb/ton).

Reduction of a suspension's pH is expected when conditioning with ferric chloride, but minimal change in pH is anticipated with cationic polymer or surfactant addition. Figure 3 shows the significant pH decreases when conditioning with ferric chloride, and confirms the negligible change in pH when conditioning with HDTMA.

We also measured SC values following conditioning, as shown in Figures 4 and 5 respectively for EBMUD and Philadelphia sludges. Comparison with Figures 1 and 2 shows that optimal CST values were invariably attained at an SC between -15 and -20. Our previous report to EBMUD also found an SC of -20 to indicate optimum dosages. This may be the level at which sufficient charge neutralization occurs to produce flocculation; since a negative offset is sometimes observed in SC readings relative to measured zeta potentials (Dentel, 1995), it is also possible that these values indicate true charge neutralization (zero zeta potential). Our comparison of SC readings with colloid titrations in a previous report to EBMUD supports this hypothesis.

Although this argues for the importance of charge neutralization as a conditioning mechanism, quite different CST values were obtained with the best dosages for each

conditioning chemical . For example, when sufficient HDTMA is added to the EBMUD sludge to give an SC of -20 (about 5000 mg/L), a CST is obtained of about 43 seconds; the ferric chloride dose giving this same SC leads to a much better CST of only 16 seconds, and the polymer dose at -20 gives a CST of 9 seconds. Thus, at equal degrees of charge neutralization, other factors must also be important in determining dewaterability.

Two doses of ferric chloride, approximately 1500 and 3000 mg/L, were selected for use in preconditioning of EBMUD sludge. These doses were at roughly 25% and 50% of the optimum dose for ferric chloride alone, and low enough to avoid significant pH change. In a similar manner, two doses of ferric chloride, approximately 1300 and 2500 mg/L, were selected for dual conditioning and dewatering of the Philadelphia sludge. In the case of HDTMA, doses of 1000 and 2000 mg/L were selected with reference to the results in Figures 1 and 2. Conditioning and CST tests were then performed varying the dose of Percol 757 that followed the ferric chloride or HDTMA.

Figures 6 and 7 show the CST results after dual chemical conditioning of EBMUD and Philadelphia sludges respectively. Figures 8 and 9 show the SC values for the same experiments. Comparison of the figures shows that a streaming current reading between -20 and -25 corresponds to the optimal dose. Unlike the comparisons between single conditioning chemicals (Figures 1 and 2), the CST values attained with the conditioner combinations all converged to similar minima (9-10 seconds). It may be hypothesized that the flocculating capability of the high-molecular weight polymer is needed to attain this result, although the polymer, ferric chloride, or surfactant is adequate to accomplish the initial degree of charge neutralization. Thus, an additive such as ferric chloride or HDTMA can be substituted for a portion of the polymer dose while still attaining the same degree of dewaterability, assuming that the CST results are indicative of centrifuge performance.

Of course, the feasibility of such a substitution depends on the actual amounts of each chemical required and the corresponding costs. Tables 2 and 3 show the pounds of Percol 757 and ferric chloride, and Percol 757 and HDTMA, required to give a CST of 10 seconds, which is assumed to correspond to satisfactory dewaterability. Doses are not given for HDTMA alone because the CST was not reduced sufficiently at any dose.

Table 2: Percol 757 and ferric chloride dose combinations required for equivalent dewaterability.

<u>Conditioner dose required for a CST of 10 sec</u> <u>(lb/ton of dry solids) *</u>				
<u>Conditioner</u>	<u>EBMUD sludge</u>		<u>Philadelphia sludge</u>	
	<u>Percol</u> <u>757</u>	<u>Ferric</u> <u>chloride</u>	<u>Percol</u> <u>757</u>	<u>Ferric</u> <u>chloride</u>
Percol 757 alone	19.8	0	14.7	0
FeCl ₃ (smaller dose) + Percol 757	13.7 (-31%)	140	12.2 (-17%)	107
FeCl ₃ (larger dose) + Percol 757	9.3 (-53%)	281	8.0 (-46%)	205
FeCl ₃ alone	0	>560	0	>425

* Numbers in parentheses are changes in Percol 757 dose due to FeCl₃ addition.

Table 3: Percol 757 and HDTMA dose combinations required for equivalent dewaterability.

<u>Conditioner dose required for a CST of 10 sec</u> <u>(lb/ton of dry solids)*</u>				
<u>Conditioner</u>	<u>EBMUD sludge</u>		<u>Philadelphia sludge</u>	
	<u>Percol 757</u>	<u>HDTMA</u>	<u>Percol 757</u>	<u>HDTMA</u>
Percol 757 alone	19.8	0	14.7	0
HDTMA (1 g/L) + Percol 757	18.6 (-6%)	93	12.2 (-17%)	81
HDTMA (2 g/L) + Percol 757	9.3 (-53%)	186	6.5 (-56%)	163

* Numbers in parentheses are changes in Percol 757 dose due to HDTMA addition.

As seen in these tables, either ferric chloride or HDTMA can reduce the Percol 757 requirement significantly. As a rule of thumb, it appears that adding a proportion of one chemical's optimum dose reduces the requirement for the other by the same amount. For example, the optimum dose of ferric chloride if used alone in conditioning the EBMUD sludge is about 560 mg/L; adding 50% of this dose reduced the polymer requirement to about 50% of its optimum dose if used alone. If this rule were invariably true, it would always be most economical to use only one of the conditioning chemicals by itself. However, the CST results also indicated that sole use of ferric chloride or HDTMA did not provide adequate dewaterability even at the optimum dose, so combined use may be desirable in some instances.

The doses in the previous tables can be converted into costs which can then be used to select an economical dose combination for a given dewatering performance. This has been done for the Percol 757 and ferric chloride dose combinations shown in Table 2, using unit costs of \$0.125 per lb for ferric chloride and \$1.71 per lb for Percol 757. These costs were obtained from suppliers in the Philadelphia area and include freight costs within a 20 mile radius of the distribution point. Table 4 shows the resulting costs per ton of dewatered solids, and Figure 10 shows total conditioner cost (Percol 757 plus ferric

chloride) as a function of the initial dose of ferric chloride. Similar calculations were not performed for HDTMA use since its cost (\$6-\$7 per pound) is prohibitive.

Table 4: Costs of Percol 757 and ferric chloride dose combinations required for equivalent dewaterability.

<u>Cost of conditioner for a CST of 10 sec</u> <u>(\$/ton of dry solids)</u>				
<u>Conditioner</u>	<u>EBMUD sludge</u>		<u>Philadelphia sludge</u>	
	<u>Percol</u> <u>757</u>	<u>Ferric</u> <u>chloride</u>	<u>Percol</u> <u>757</u>	<u>Ferric</u> <u>chloride</u>
Percol 757 alone	33.8	0	25.0	0
FeCl ₃ (smaller dose) + Percol 757	23.4	17.5	20.9	13.3
FeCl ₃ (larger dose) + Percol 757	15.9	35.1	13.7	25.6

Although preconditioning with ferric chloride reduced the cationic polymer requirement, these results indicate that the process was not cost effective. The same conclusion is likely to apply in general since the relative costs of ferric chloride and the polymer would have to change substantially to alter the trends shown in Figure 10. However, if the addition of ferric chloride to anaerobically digested biosolids is considered for other purposes, the anticipated reduction in polymer usage should be considered in cost calculations.

Conclusions

Cationic polymer Percol 757, cationic surfactant HDTMA, and ferric chloride each brought about significant reductions in the CST of anaerobically digested sludge from two different sites. The polymer provided the lowest CST and required a much lower dose than did the other conditioners.

The optimum dose of each conditioner occurred when the streaming current detector's reading was between -15 and -20. Although this degree of charge neutralization seemed necessary for good flocculation, it was not sufficient to guarantee a very low CST; for this, some amount of polymer was needed beyond the ferric chloride or HDTMA dose. The polymer enhanced dewaterability by some mechanism--such as bridging--that was in addition to charge neutralization.

Use of dual chemical conditioners reduced the required dose of either chemical as used separately. Adding a proportion of one chemical's optimum dose reduced the requirement for the other by roughly the same amount.

The use of ferric chloride or HDTMA as a preconditioner can reduce the polymer requirement, this is not a cost effective option at current prices for these additives.

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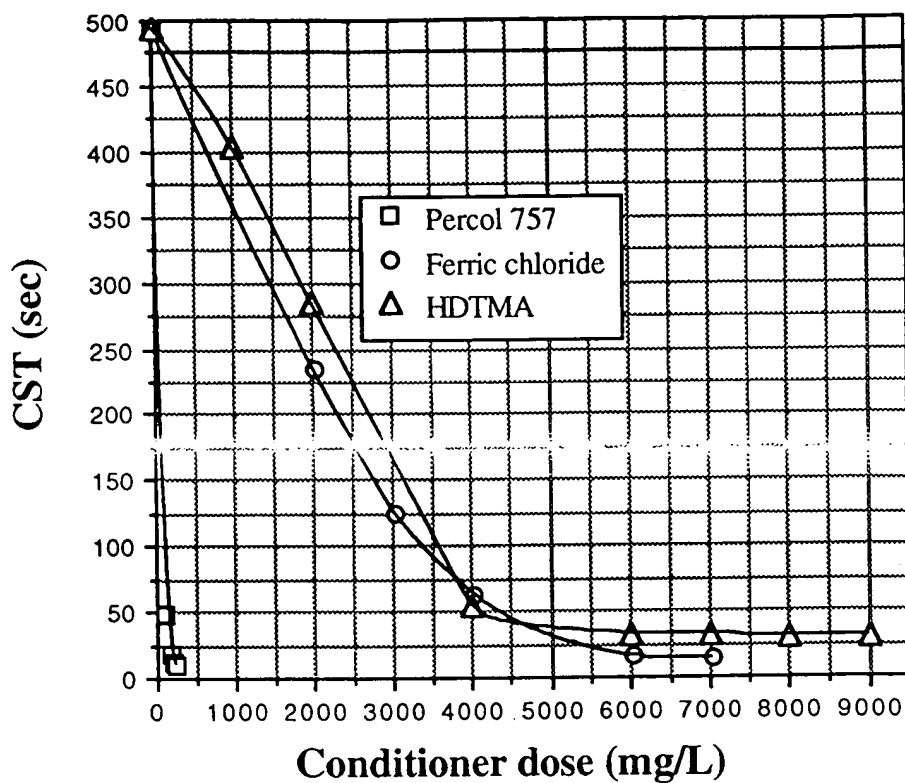


Figure 1. CST results after conditioning of EBMUD sludge.

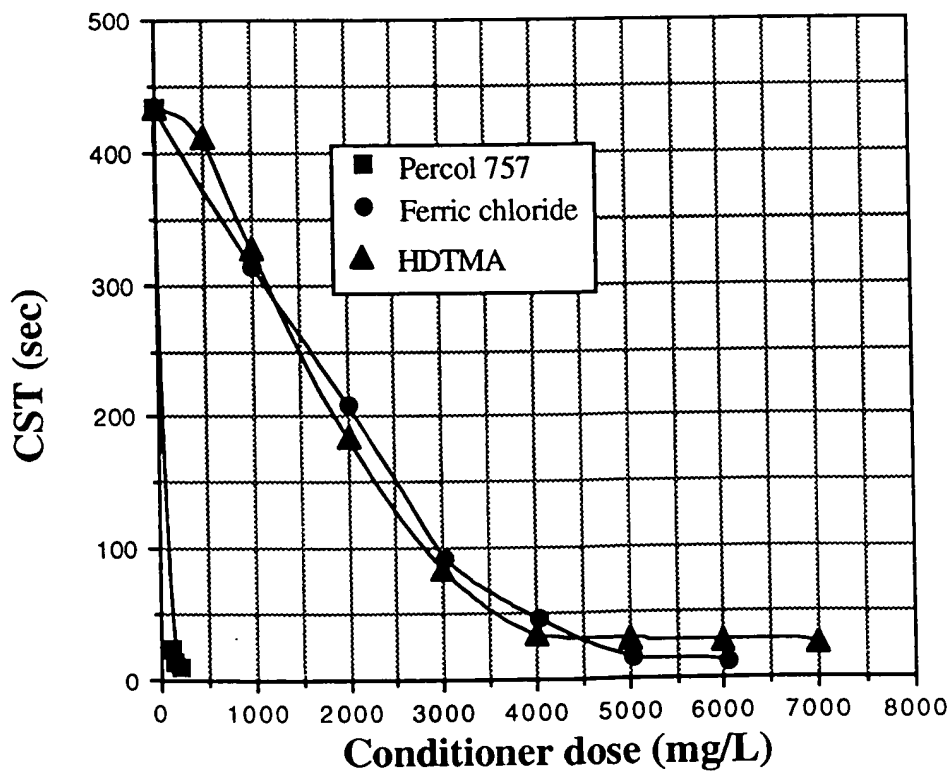


Figure 2. CST results after conditioning of Philadelphia sludge.

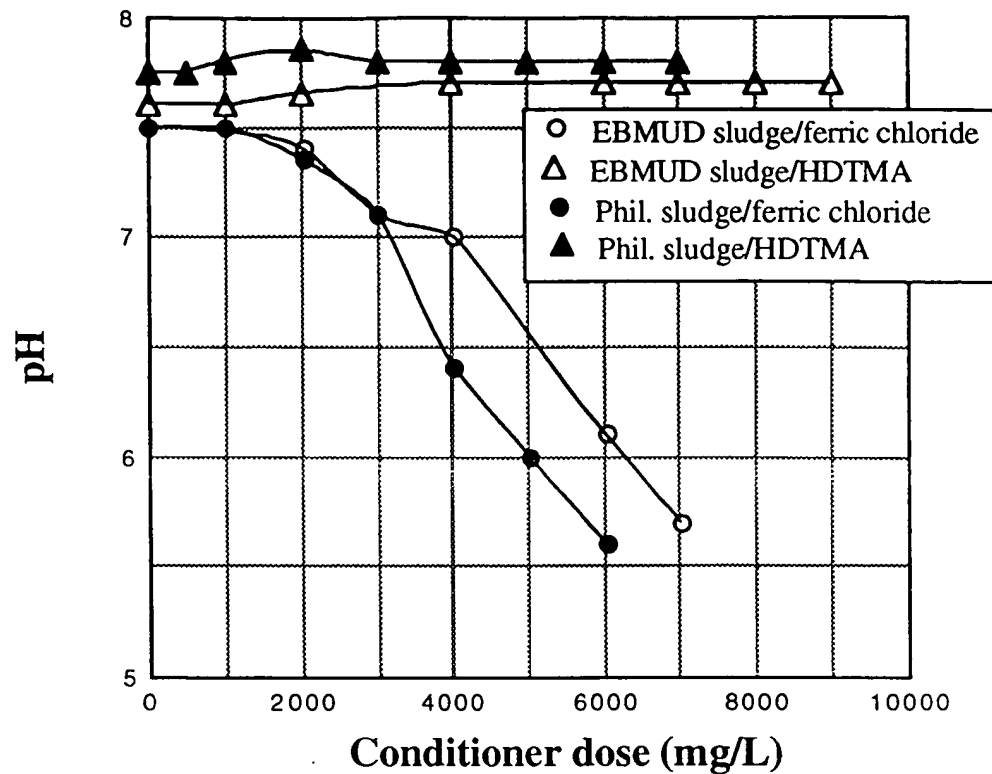


Figure 3. pH values after conditioning of EBMUD and Philadelphia sludges.

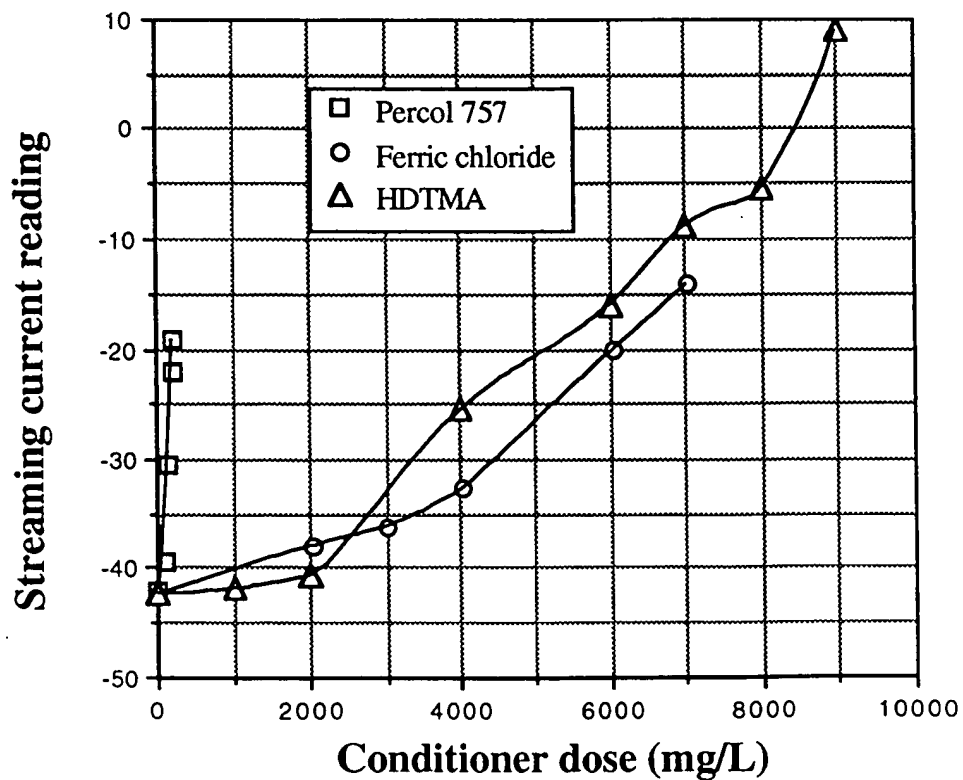


Figure 4. Streaming current results after conditioning of EBMUD sludge.

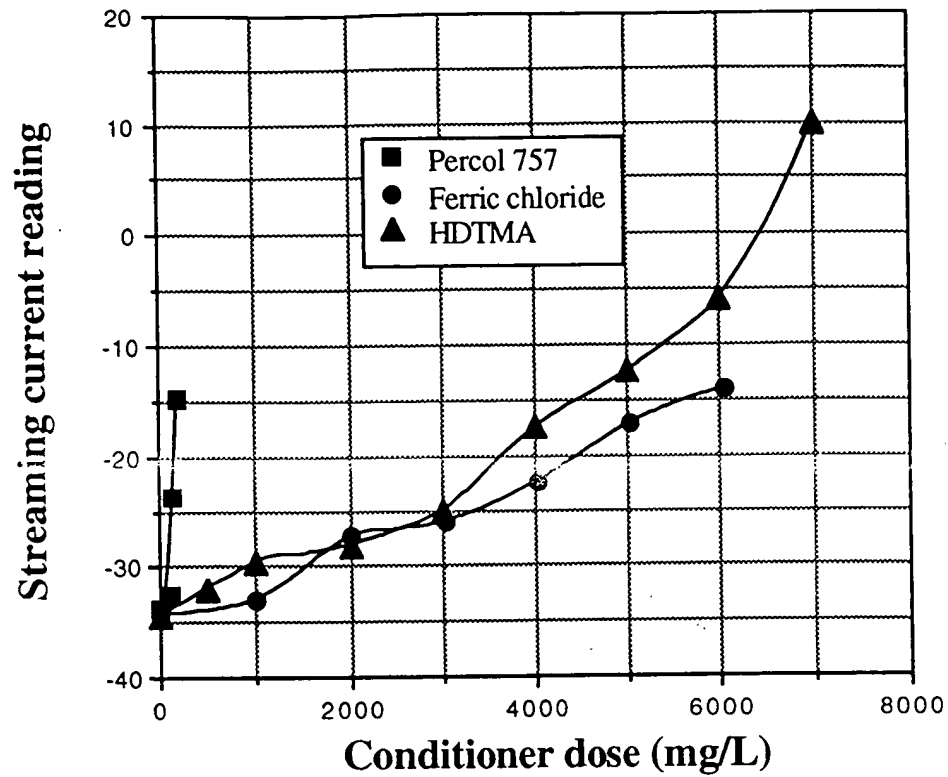


Figure 5. Streaming current results after conditioning of Philadelphia sludge.

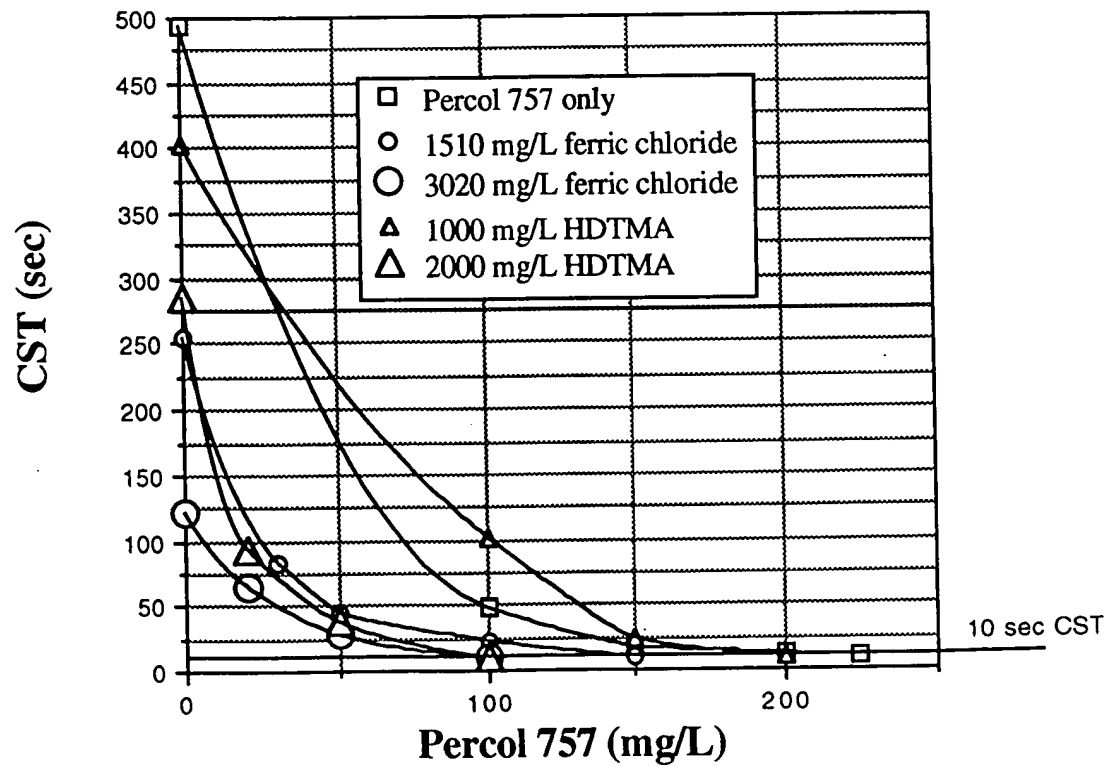


Figure 6. CST results after dual chemical conditioning of EBMUD sludge.

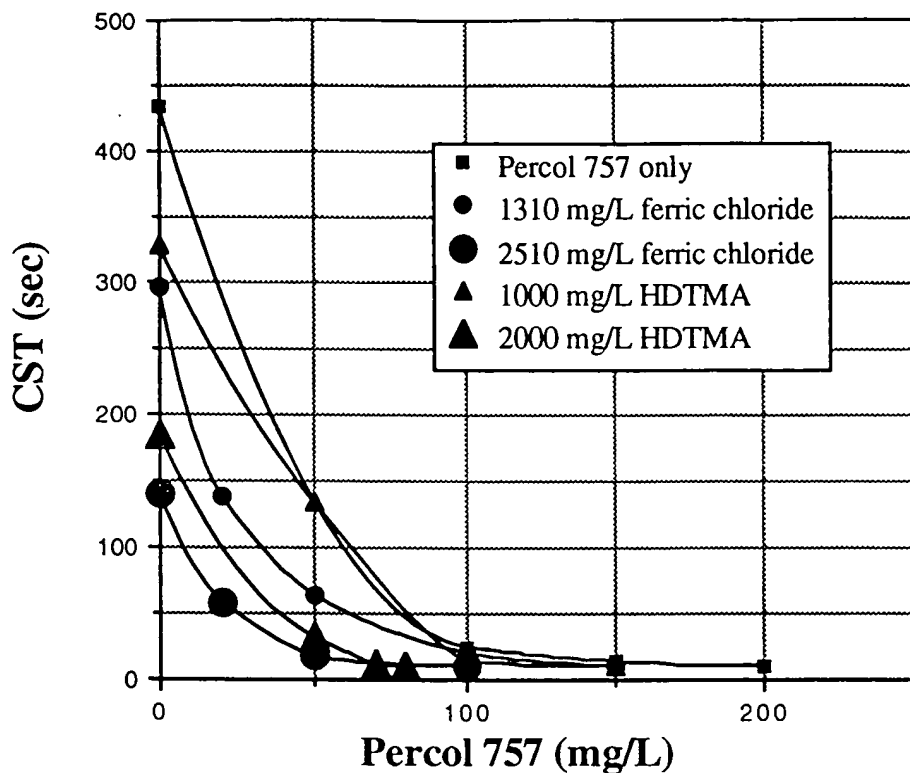


Figure 7. CST results after dual chemical conditioning of Philadelphia sludge.

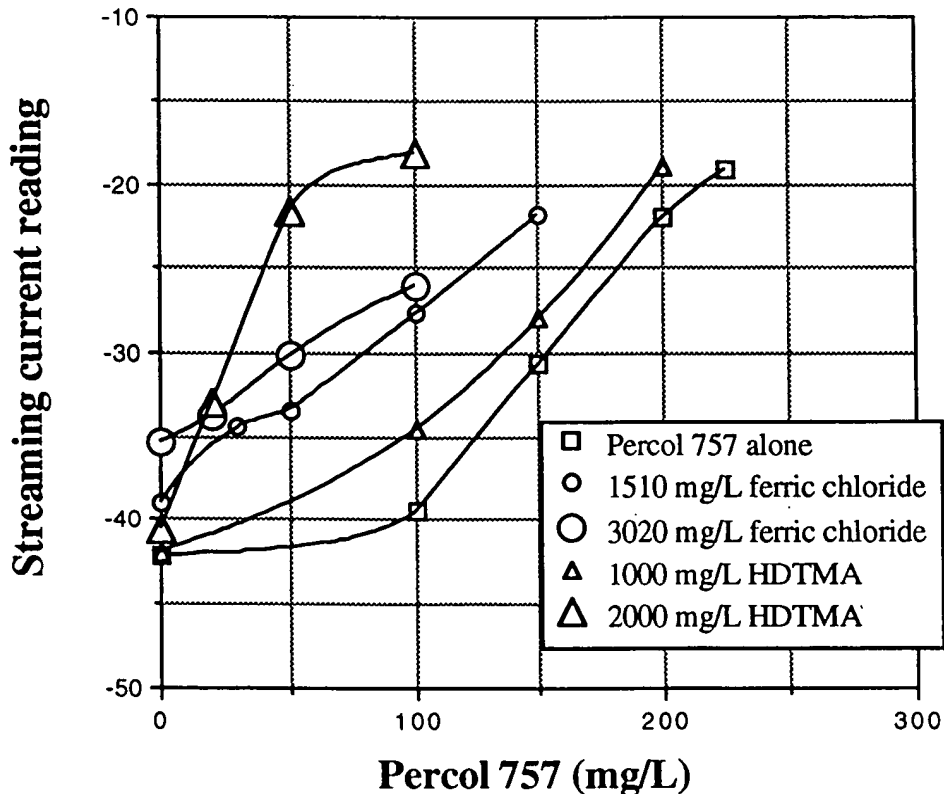


Figure 8. Streaming current results after dual chemical conditioning of EBMUD sludge.

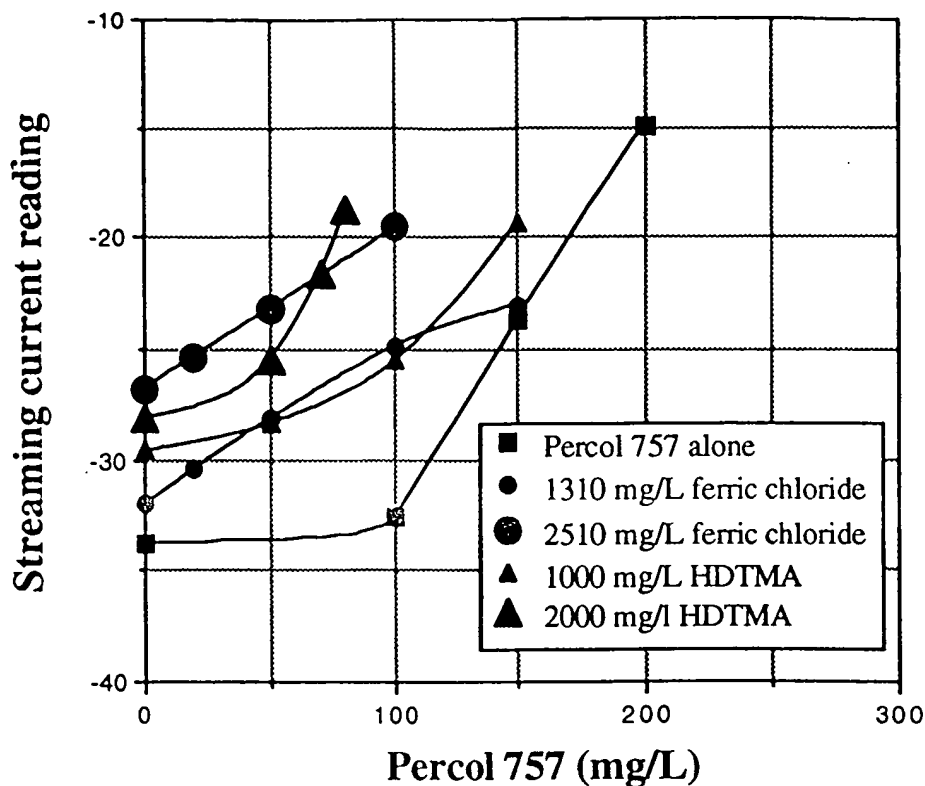


Figure 9. Streaming current results after dual chemical conditioning of Philadelphia sludge.

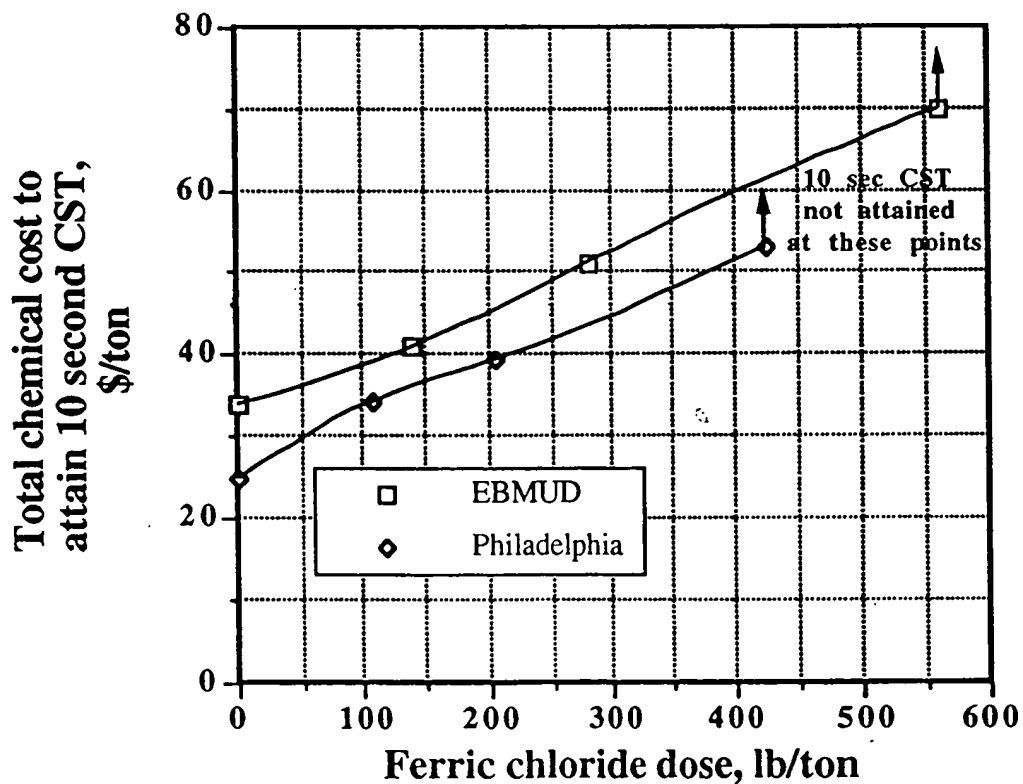


Figure 10. Equivalent chemical costs varying initial ferric chloride dose

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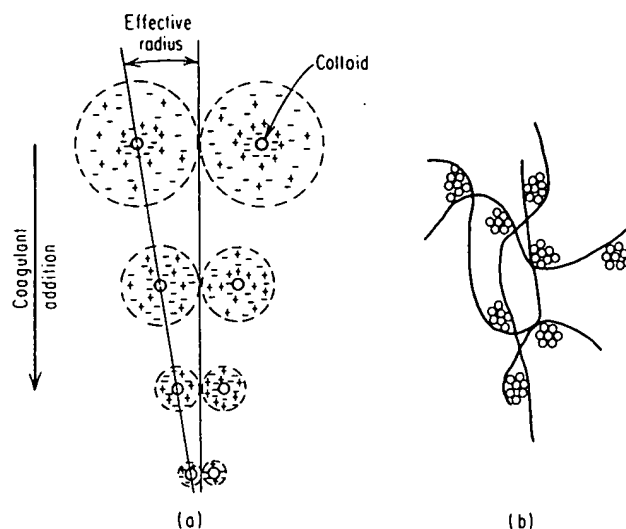


FIG. 8.1 (a) *Coagulation*: The addition of a coagulant neutralizes charges, collapsing the "cloud" surrounding the colloids so they can agglomerate. (b) *Flocculation*: The bridging of the flocculant chemical between agglomerated colloidal particles forms large settleable flocs.

an effective size and settling rate; but even larger particles, which are not truly colloidal and would settle if given enough time, may require coagulation to form larger, faster settling floc.

When insufficient settling time is available in a treatment plant to remove suspended solids, coagulation and flocculation may cause them to grow in size and settle rapidly enough to overcome the physical limitation of the plant design.

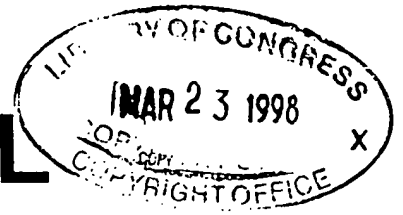
Colloids are categorized as hydrophobic (water hating) or hydrophilic (water loving). Hydrophobic colloids do not react with water; most natural clays are hydrophobic. Hydrophilic colloids react with water; the organics causing color are hydrophilic. Of importance in water treatment, the hydrophilic colloids may

TABLE 8.1 Sedimentation of Small Particles of Silica of 2.65 Sp. Gr.

Typical	mm	μm	Surface area (total)	Settling time, 1 m fall
Gravel	10.	10,000	3.14 cm^2	1 s
Coarse sand	1.	1,000	31.4 cm^2	10 s
Fine sand	0.1	100	314 cm^2	125 s
Silt	0.01	10	0.314 m^2	108 min
Bacteria	0.001	1.	3.14 m^2	180 hr
Colloidal matter	0.0001	0.1	31.4 m^2	755 days

NOTE: Particles larger than 100 μm are visible to the naked eye and are considered to be settleable solids. In the range of 10–100 μm , they are considered to be turbid. Below 10 μm they are considered colloidal. Particles larger than 0.1 μm are visible by light microscope; below 0.1 μm , the electron microscope is used for detection.

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EVALUATION OF DUAL CHEMICAL CONDITIONING AND DEWATERING OF ANAEROBICALLY DIGESTED BIOSOLIDS

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INTRODUCTION

The optimal chemical conditioning and dewatering of a municipal sludge is a challenging task. This is due to the complex and changeable physical/chemical makeup of these materials. The means by which chemical conditioners interact with the colloidal phase in biological suspensions to facilitate the release of water is poorly understood, with the optimal amounts and types of conditioners required depending on a variety of factors. These include both aqueous and surface chemistries of the sludge, and the physical properties of the suspended solids. Also important is the chemistry of any chemical conditioner used, and how it interacts with the biosolids. The success of any conditioning process will also depend on the specific dewatering process employed. Thus, the sludge conditioning process is a multivariate problem with no simple strategy available for its optimization. At present, the required dosages for chemical conditioners must be determined empirically. With this being the case, the use of multiple chemical additives becomes less feasible because of the difficulty in identifying a proper dose combination. Nonetheless, it is possible that a combination of conditioners might enhance water removal, solids capture, and solids yield for some applications.

This work (Deniel and Chitkela, 1993) investigated the combined use of an inorganic conditioner (ferric chloride) or a cationic surfactant (HDTMA, i.e., hexadecyltrimethylammonium bromide), and an organic polyelectrolyte (Pentrol 757A) for the conditioning and dewatering of municipal anaerobically digested sludges. In the past, ferric chloride was more commonly used in conjunction with lime, but the current practice is generally to use cationic polymers (polyelectrolytes) alone. The inorganic conditioners require doses of up to 20 percent on a dry solids basis and typically cannot produce the solids concentrations in dewatered biosolids that are attainable with much lower dosages of polymer (approximately 1 percent on a dry solids basis). In addition, the use of cationic surfactants in sludge conditioning has not been previously reported, and we wished to examine the ability of such an additive to condition by charge neutralization alone.

BACKGROUND

This study was initiated by the East Bay Municipal Utility District (EBMUD) in Oakland, California due to its high polymer usage in conditioning anaerobically digested sludge (ADS). For such applications the EPA (1987) indicates a usual dosage range of 1.5-7.5 kg/Mg (3-15 lb/ton) with a typical value of 3.0 kg/Mg (6 lb/ton). The required dose at this facility was approximately 17 kg/Mg (34 lb/ton) of dry solids. Previous laboratory comparisons based on capillary suction time (CST) had confirmed the necessity for this high dose; for example, to attain the CST resulting from a 7.5 kg/Mg (15 lb/ton) polymer dose to an ADS from Wilmington, Delaware, the EBMUD biosolids required 17 kg/Mg (34 lb/ton) of the same polymer. Both electrical charge and rheological properties of the sludges were indicated as contributing factors (Deniel et al., 1994).

The EBMUD sludge was shown to have a more negative zeta potential and a greater concentration of negative charge than the Wilmington sludge. The conditioning mechanisms attributed to ferric chloride include the formation of positively charged iron hydroxide precipitate, which can then neutralize the negatively charged sludge solids. It thus appeared plausible that this conditioner could be substituted for

a more expensive polyelectrolyte in accomplishing a portion of the required charge neutralization. The cationic organic polymer might still be needed to bridge the particles and make the sludges sufficiently dewaterable. The detailed principles of inorganic chemical and organic polymer conditioning of sludges are available elsewhere (EPA, 1987; Gregory, 1993; Abu-Ofi, 1994). The literature on dual chemical conditioning of sludges is scarce, although one study conducted by Roberts and Olsson (1975) showed that an activated sludge (at 0.7 percent solids) that needed a polyelectrolyte dose of 3.57 kg/Mg (7.14 lb/ton) of dry solids to attain a CST of 18 seconds required only 1.4 kg/Mg (2.86 lb/ton) to reach the same 18 second CST when pre-conditioned with a 0.5 g/L dose of alum. Although dual chemical conditioning reduced the polyelectrolyte requirement by 60 percent, the resulting alum requirement was very high, i.e., 71.65 kg/Mg (143 lb/ton) of dry solids.

This work evaluated a two-stage conditioning of anaerobically digested sludges consisting of initial addition of the ferric chloride or the HDTMA followed by use of a cationic polymer, Percol 757. Using capillary suction time (CST) as a laboratory indicator of dewaterability, combined doses were to be identified that produced CST values equal to that obtained with polymer alone. In order to ascertain whether charge neutralization is mechanistically important in this manner, we utilized a streaming current detector to observe trends in colloidal charge occurring with the use of these conditioners. An economic evaluation of dual conditioner use was to be completed to determine whether a cost reduction could be effected by partial use of the less expensive ferric chloride, or the HDTMA.

EXPERIMENTAL PROCEDURES

Materials: In addition to the biosolids from the EBMUD facility, anaerobically digested biosolids were also obtained from Philadelphia's Biosolids Recycling Center for comparison purposes. Ferric chloride (formula weight 162.21) was obtained from Fisher Scientific (Malvern, PA.) and prepared as a 5.029 percent solution for use. The cationic surfactant used as a conditioner was hexadecyltrimethyl ammonium bromide (HDTMA), a quaternary amine cationic surfactant with a formula weight of 364.5. Also obtained from Fisher, it was prepared and used as a 4 percent solution. Percol 757 was used as a representative cationic polymer. A product of Allied Colloids, Inc., it is supplied as a dry copolymer of acrylamide and quaternized dimethylaminoethylacrylate at a mole ratio of approximately 60%/40%. It was obtained after preparation to solution form in the full-scale facilities of the Wilmington (Delaware) Water Pollution Control Facility, where it is used for sludge conditioning. The solution was determined to have a total solids content of 0.50 percent. A syringe was used to dispense this solution into biosolids samples. A household blending mixer (Braun Multipractic) and a Phipps & Bird jar test apparatus with standard 1"x3" paddles were used for mixing of sludges in the conditioning process. A capillary suction time (CST) instrument (Venture Innovations) was used for dewaterability assessments. A streaming current detector (Milton Roy model SCA200) was used for determination of relative colloidal charge. The pH meter used in this work was an Accumet® model 830 (Fisher Scientific).

Methods: i) Conditioning of sludges was conducted using the general procedure given in the Guidance Manual for Selection of Polymers in Wastewater Treatment Plants (WEIRF, 1993). ii) Conditioning and dewatering of sludges by Percol 757, ferric chloride, or HDTMA alone: A representative 0.5 L volume of each sludge was placed in a 1.0 L beaker and rapidly mixed with the household blending mixer for 6-7 seconds after addition of the specified dose of Percol 757, ferric chloride, or HDTMA. Then using the jar test apparatus, rapid mixing was continued for 20 seconds at approximately 117 RPM, and finally, the suspension was flocculated at approximately 28 RPM for 2 minutes. The suspension's pH, CST and streaming current (SC) values were then measured. All experimental values presented in this report are averages of at least three analyses. iii) Conditioning and dewatering of sludges by ferric chloride or HDTMA followed by Percol 757: The same procedure was used except that samples were rapidly mixed for 5 seconds after addition of the specified dose of ferric chloride or HDTMA, and for an additional 5 seconds following the specified dose of Percol 757.

The above mixing conditions were selected to simulate as closely as possible the flow conditions (velocity gradients and mixing times) from the point of chemical addition up to the centrifuges at the EBMUD facility. The Gt value, calculated by summing products of velocity gradients (G) and respective residence times (t), was between 7300 and 9500 depending on the sludge viscosity (which is dependent in turn on polymer dose). This range has been reported as optimal in previous studies of ADS conditioning (Novak 1989, 1990).

RESULTS AND DISCUSSION

Comparison of sludge properties. A comparison of EBMUD and Philadelphia sludge pH values showed no major differences. The Philadelphia sample had a slightly greater total solids content, by a factor of 14%. Although a 14 percent higher solids concentration was observed in Philadelphia sludge, the CST and original SC values for EBMUD were nearly 14 percent and 23.5 percent higher in magnitude, respectively. This shows that EBMUD would need more cationic polymer in sludge conditioning than the Philadelphia sludge.

Comparison of results of single conditioner dosages. Figure 1 provides the results of conditioning and dewatering for the EBMUD sludge, when conditioned with Percol 757, ferric chloride, or HDTMA individually. These results once again confirm that cationic polymers are very effective in sludge conditioning when compared to the inorganic chemical conditioning with ferric chloride. Use of a cationic surfactant in sludge conditioning is unconventional, but its optimum dose on a mass basis was comparable to that of ferric chloride. However, this finding is of little practical use since HDTMA is significantly more expensive than ferric chloride or flocculant polymers per unit mass. Cationic surfactants such as HDTMA are also used as germicides, herbicides, fabric softeners, and corrosion inhibitors (Rosen, 1989) and thus would require evaluation with respect to other impacts as well. These results also showed a greater conditioner requirement for the EBMUD biosolids than for the Philadelphia biosolids (results on Philadelphia solids are not shown in this paper). To obtain a CST of approximately 10 seconds, the required Percol 757 additions for EBMUD and Philadelphia respectively were 0.215 and 0.180 g/L or, on a mass per mass basis, 10 and 7.5 kg/Mg (i.e., 1% and 0.75% respectively). The ferric chloride additions never reduced the CST to 10 seconds for either sample, but to provide a CST of 15 seconds the doses were 6.0 and 5.2 g/L respectively (280 and 215 kg/Mg and which are 28% and 21.5% respectively). In the case of HDTMA, at a dose of 7.0 g/L the final CSTs of both EBMUD and Philadelphia sludges were reduced to 32 and 27 seconds respectively. Since cationic surfactants are very expensive, i.e., \$13-\$15 per kg, the other dose estimations were not conducted.

Reduction of a suspension's pH is expected when conditioning with ferric chloride, but minimal change in pH is anticipated with cationic polymer or surfactant addition. Results showed significant pH decreases when conditioning with ferric chloride, and confirms the negligible change in pH when conditioning with HDTMA. We also measured SC values following conditioning for EBMUD and Philadelphia sludges. Comparison of the results provided that optimal CST values were invariably attained at an SC between -15 and -20. This may be the level at which sufficient charge neutralization occurs to produce flocculation; since a negative offset is sometimes observed in SC readings relative to measured zeta potentials (Dentel, 1995), it is also possible that these values indicate true charge neutralization (zero zeta potential). Comparison of SC readings with colloid titrations in a previous report to EBMUD supports this hypothesis (Dentel et al., 1994). Although this argues for the importance of charge neutralization as a conditioning mechanism, quite different CST values were obtained with the best dosages for each conditioning chemical. For example, when sufficient HDTMA is added to the EBMUD sludge to give an SC of -20 (about 5.0 g/L), a CST is obtained of about 43 seconds; the ferric chloride dose giving this same SC leads to a much better CST of only 16 seconds, and the polymer dose at -20 gives a CST of 9 seconds. Thus, at equal degrees of charge neutralization, other factors must also be important in determining dewaterability.

Evaluation elements of dual conditioning application. Two doses of ferric chloride, approximately 1.5 g/L (the natural dosage) and 3.0 g/L (the larger dose), were selected for use in preconditioning of EBMUD sludge. These dosages were at roughly 5% and 50% of the optimum dose for ferric chloride alone, and low enough to avoid significant pH change. In a similar manner, two doses of ferric chloride, approximately 0.5 g/L (the smaller dose) and 2.5 g/L (the larger dose), were selected for dual conditioning and dewatering of the Philadelphia sludge. Conditioning and CST tests were then performed varying the dose of Percol 757 and HDTMA followed the ferric chloride or HDTMA. Figure 2 shows the CST results after dual chemical conditioning of EBMUD sludge. The SC values for the same experiments are presented in Figure 3A. Comparison of these results showed that a streaming current reading between -20 and -25 corresponded to the optimum dose of the conditioner. Unlike the comparisons between single conditioning chemicals (Figure 1), the CST values obtained with the conditioner combinations all converged to similar values (9-20 seconds). It may be hypothesized that the flocculating capability of the high-molecular-weight polyelectrolyte is needed to attain the result, although the polymer, ferric chloride, or surfactant is adequate to accomplish the initial dose of charge neutralization. Thus, an additive such as ferric chloride or HDTMA can be substituted for a portion of the polymer dose while still attaining the same degree of dewaterability of the sludge.

Of course, the desirability of such a substitution depends on the actual amounts of each chemical required and the corresponding costs. Tables 1 and 2 show the pounds of Percol 757 and ferric chloride, and Percol 757 and HDTMA, required to give a CST of 10 seconds, which is assumed to correspond to satisfactory dewaterability. Doses are sufficient to give a CST of 10 seconds. As seen in the tables, either ferric chloride or HDTMA can reduce the Percol 757 requirement significantly. As a rule of thumb, it appears that adding a proportion of one chemical's optimum dose reduces the requirement for the other by the same amount. For example, the optimum dose of ferric chloride if used alone in this case reduced the polymer requirement to about 50% of its optimum dose if used alone. If this rule were generally true, it would always indicate that sole use of ferric chloride or HDTMA did not provide adequate dewaterability even at the optimum dose, so combined use may be desirable in some instances.

The dosages in the previous tables can be converted into costs which can then be used to select an economical dose combination for a given dewatering performance. This has been done for the Percol 757 and ferric chloride dose combinations shown in Table 1, using unit costs of \$0.27 per lb for ferric chloride and \$1.77 per lb for Percol 757. These costs were obtained from suppliers in the Philadelphia area and include freight costs within a 20 mile radius of the distribution point. Table 3 shows the resulting costs per acre-foot dewatered solids, and Figure 4 shows total conditioner cost (Percol 757 plus ferric chloride) as a function of the initial dose of ferric chloride. Similar calculations were not performed for HDTMA, because its cost is prohibitive. Although preconditioning with ferric chloride reduced the cationic polymer requirement, these results indicate that the process was not cost effective. The same conclusion is likely to hold in general since the relative costs of ferric chloride and the polymer would have to change substantially to alter the trend shown in Figure 4. However, if the addition of ferric chloride to anaerobically digested biosolids is considered for other purposes, such as odor and struvite control, the anticipated reduction in polymer usage should be considered in cost calculations.

CONCLUSIONS

Cationic polymer Percol 757, cationic surfactant HDTMA, and ferric chloride each brought about significant reductions in the CST of anaerobically digested sludge from two different sites. The polymer provided the lowest CST and required a much lower dose than did the other conditioners. The optimum dose of each conditioner occurred within the streaming current detector's reading was between -15 and -20. Although the degree of charge neutralization seemed necessary for good flocculation, it was not sufficient to guarantee a very low CST for this, some amount of polymer was needed beyond the ferric

chloride or HDTMA dose. The polymer enhanced dewaterability by some mechanism--such as bridging--that was in addition to charge neutralization. Use of dual chemical conditioners reduced dose of either chemical as used separately. Adding a proportion of one chemical's optimum dose reduced the requirement for the other by roughly the same amount. The use of ferric chloride or HDTMA as a preconditioner can reduce the polymer requirement, but this is not a cost effective option at current prices for these additives.

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Table 1. Percol 757 and ferric chloride dose combinations required for equivalent dewaterability.

Conditioner	Conditioner dose required for a CST of 10 sec (kg/kg of dry solids)		
	EBMUD sludge Percol 757	Ferric chloride	Philadelphia sludge Percol 757 Ferric chloride
Percol 757 alone	9.9	0	7.4
FeCl ₃ (smaller dose) + Percol 757	6.8 (-31%)	70.0	6.1 (-17%)
FeCl ₃ (larger dose) + Percol 757	4.6 (-53%)	140.5	4.0 (-46%)
FeCl ₃ alone	0	>280	0
			>212.5

*Numbers in parentheses are changes in Percol 757 dose due to FeCl₃ addition.

Table 2. Percol 757 and HDTMA dose combinations required for equivalent dewaterability.

Conditioner	Conditioner dose required for a CST of 10 sec (kg/kg of dry solids)		
	EBMUD sludge Percol 757	Philadelphia sludge HDTMA	HDTMA
Percol 757 alone	9.9	0	7.4
HDTMA (1 g/L) + Percol 757	9.3 (-6%)	46.5	6.1 (-17%)
HDTMA (2 g/L) + Percol 757	4.7 (-53%)	93.0	3.3 (-56%)

*Numbers in parentheses are changes in Percol 757 dose due to HDTMA addition.

Table 3. Costs and Percol 757 and ferric chloride dose combinations required for equivalent dewaterability.

Conditioner	Cost of conditioner for a CST of 10 sec (\$/mg of dry solids)		
	EBMUD sludge Percol 757	Philadelphia sludge Percol 757	Philadelphia sludge Ferric chloride
Percol 757 alone	37.3	0	27.6
FeCl ₃ (smaller dose) + Percol 757	25.8	19.3	23.0
FeCl ₃ (larger dose) + Percol 757	17.5	38.7	15.1

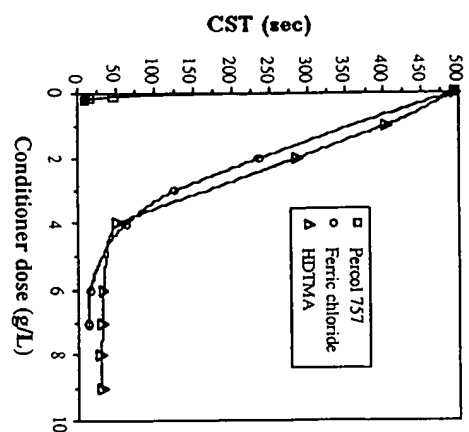


Figure 1. CST results after conditioning of EBMUD sludge.

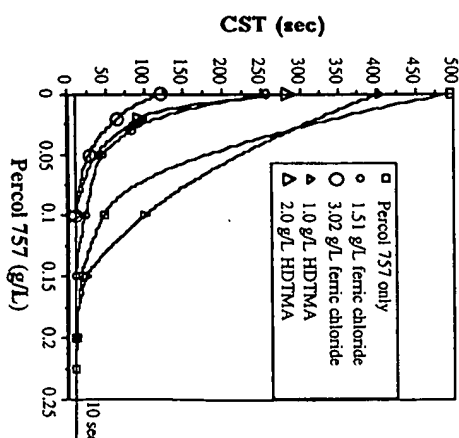


Figure 2. CST results after dual chemical conditioning of EBMUD sludge.

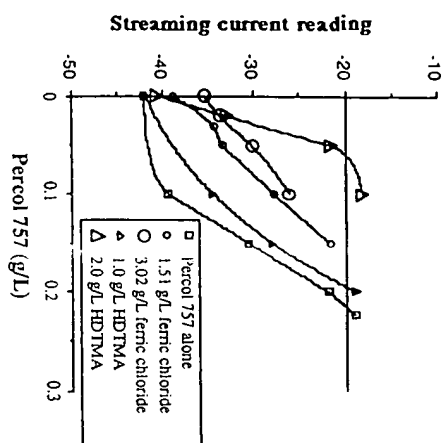


Figure 3. Streaming current results after dual chemical conditioning of EBMUD sludge.

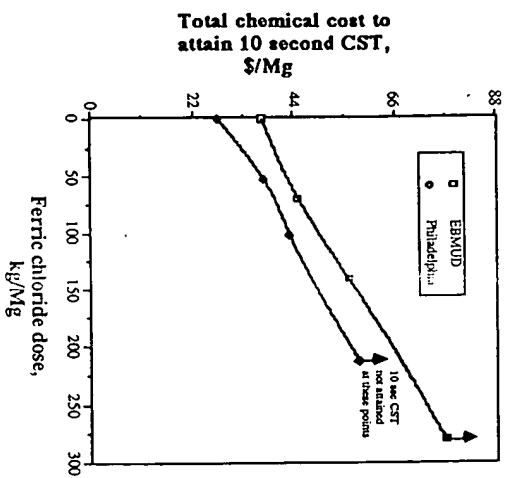


Figure 4. Equivalent chemical costs varying initial ferric chloride dose.